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16. ABSTRACT

This literature review summarizes current data on the effects of pesticides and trace contaminants on marine and estuarine organisms. Impacts of the petroleum industry on marine ecosystems is reviewed from several viewpoints including world input to the oceans and French coast, biotransformation, oil spill technology, hydrocarbon characterization, toxicity, blowouts, and liability.

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Marine and estuarine pollution

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The results of two international marine pollution symposia held in 1979 were published.[1,2] Topics included heavy metal pollution, radioactive materials, oil and related substances, pesticides, industrial and domestic pollution, environmental evaluation and management, biochemical, physiological, genetic, and behavioral effects, pathobiology, and monitoring. Workshops were conducted[2] that outlined current knowledge in the field and suggestions for future research.

The results of the offshore ecological investigation of 1972–1974, which studied the long-term effects of oil well drilling and production off Louisiana, were published with an independent appraisal of the study.[3] The general conclusion was that no long-term effects of oil drilling and production could be detected in this area and natural phenomena probably have a greater impact on the ecosystem than petroleum activity.

PESTICIDES

The U. S. Department of Commerce published three annotated bibliographies on the toxic effects of pesticides on non-target organisms including marine species.[4-6] Zell and Ballschmiter[7] discussed the global occurrence of hexachlorobenzene (HCB) and polychlorocamphenes (toxaphene) in fish from pristine environments throughout the world. In a review of the effects of pollution on estuarine clams and snails, Menzel[8] discussed the toxicity of pesticides to various mollusks.

Williams and Duke[9] provided an overview of the knowledge of the effects of chlorinated hydrocarbons, organophosphates, carbamates, and juvenile hormone mimics on crabs. Highlighted were (1) the effect of Kepone on crabs in the James River and (2) the impact of mirex on estuarine crabs following its use to control fire ants in the southeastern U. S.

Methods. A marine algal bioassay method suitable for conducting short- and long-term studies on pesticides and industrial wastes was described.[10] Algae were grown in optically matched culture tubes that fit directly into a spectrophotometer, allowing population density to be estimated by absorbance without removal of samples. Hamburger[11] discussed the problems associated with the conduct of the bioaccumulation test required by Japanese legislation and made suggestions for improvement.

Hemmer[12] described a positive pressure proportional diluter that can be employed in toxicological studies of pesticide effects on estuarine organisms. A bioassay method developed for the screening of pyrethroids for contact toxicity against the wood-boring marine crustacean, *Limnoria*, was discussed.[13] This method is suitable for use with insecticides with low water solubility.

Stout and Beezhold[14] described simple methods for extraction and clean-up of marine fish and fish product samples for chlorinated hydrocarbons. Yamagishi *et al.*[15] discussed an electron capture gas chromatographic technique for determining 1,3-Dichloro- and 1,3,5-Trichloro-2-(4-nitrophenoxy) benzine (NIP and CNP) in estuarine fish and shellfish.

Acute effects. McLeese and Metcalfe[16] found that the joint toxicity of mixtures of phosphamidon and methidathion to adult lobsters (*Homarus americanus*) was greater than additive in tests with the combined 48-hr toxicity unit of the mixtures totaling 0.25, 0.5, and 1.0 units. The largest magnification factor for greater than additive toxicity occurred where phosphamidon and methidathion contributed to the total toxicity unit in a ratio 1:1:5 and was progressively smaller for ratios of 1:4:4 and 2:1.

Hirayama and Tamanoi[17] found that for *Penaeus japonicus* the 24-h TL_m of MEP at 25°C decreased with larval development from 1.9 mg/l at nauplius stage to $5.0 - 9.0 \times 10^{-4}$ mg/l at post larval stage; for diazinon from over 20 mg/l at the nauplius stage to $2 - 5 \times 10^{-2}$ mg/l at post larval stage. For *Portunus trituberculatus*, the 24-h TL_m value also decreased with development, the megalop stage having the lowest values of $1.7 - 5.0 \times 10^{-4}$ mg/l MEP and $4.0 - 15.0 \times 10^{-3}$ mg/l diazinon.

Diflubenzuron was not acutely toxic to shellfish and fish, but affected the survival rate of *Artemia salina* and the shrimp *Palaemonetes varians* when its concentration rose to its level of water solubility.[18] Linden *et al.*[19] determined the toxicity of 28 pesticide formulations to the bleak *Alburnus alburnus* and the harpacticoid *Nitocra spinipes*, two brackish water organisms.

Rubinstein *et al.*[20] used a multi-species exposure system to determine the effects of contaminated sediments on estuarine organisms and settling communities. Oyster shell growth on artificially prepared Kepone-contaminated sediment was significantly inhibited at 10 and 1 mg/l Kepone, but no mortality was observed. After a 10-d exposure to 10 mg/l Kepone, 37% of the mysids had died. Substrate reworking by lugworms was completely inhibited at 10 mg/l Kepone although no mortality occurred. After 28 d at 10 mg/l, oysters had accumulated 1.2 mg/l and lugworms 1.1 mg/l. Of the microfauna harvested at 28 d representing 19 species from 4 phyla, polychaetes, the only group affected, were almost completely eliminated at 10 mg/l Kepone. When James River sediment containing Kepone was tested, it was found that mysid mortality was insignificant but oyster shell growth and lugworm reworking were significantly inhibited. After a 28-d exposure, oysters contained 0.51 mg/l Kepone and lugworms 0.75 mg/l. Additional pesticide acute effects information is presented in Table 1.[21-22]

Sublethal effects. Weis and Weis[23] found that carbaryl and parathion induced cardiovascular abnormalities and developmental arrest in killifish (*Fundulus heteroclitus*), whereas DDT and malathion had no effect up to 10 mg/l. DDT, malathion, and carbaryl caused optic abnormalities in the Atlantic silver-side (*Menidia menidia*) at concentrations as low as 0.01 mg/l. Sheepshead minnows (*Cyprinodon variegatus*) did not exhibit abnormalities with any insecticide, but malathion caused uncoordinated neuromuscular activity in newly hatched fry.

Bookhout *et al.*[24] observed differential survival of the mud-crab, *Rhithropanopeus harrisii*, from hatching to first crab stage in 35–125 mg/l Kepone, whereas differential survival of the blue crab, *Callinectes sapidus*, over the same period of development, occurred in a range of 0.1 to 1.0 mg/l.

Using developing estuarine macrobenthic communities, Tagatz *et al.*[25] found that significantly fewer chordates, mollusks, and annelids occurred in 819 mg/l of the drilling mud biocide SURFLO B33 than in the control. Average numbers of animals and species did not significantly decrease in the biocide ALDACIDE at 15 and 300 mg/l. The authors concluded that ALDACIDE, which contains paraformaldehyde, was less toxic than the SURFLO B33, which contains dichlorophenol and showed toxic characteristics similar to those of PCP. Cook *et al.*[26] examined the effects of pesticides on the microfungal succession in estuarine microcosms. Resident fungi were succeeded by *Fusarium* in the carbaryl-treated microcosm; by a fungus in the *Penicillium chrysogenum* series in the methyl parathion-treated microcosm; and by a fungus in the *Penicillium canescens* series in the pentachloro-phenol-treated microcosm.

Metabolism. Ghiasuddin and Matsumura[27] used DDT to inhibit the enzyme ecto-ATPase in the nerve cord of the walking leg of the American lobster *Homarus americanus*. DDT inhibited the maintenance of a proper concentration of surface Ca^{++} in the nerve membrane, which may contribute to the unstable and excited condition of DDT-poisoned axons.

AChE activity of an isolated nerve cord of moribund pink shrimp (*Penaeus duorarum*) was significantly depressed by exposure for 6 h to 1.3 mg/l methyl parathion.[28] Methyl paraoxon did not depress AChE activity after exposure for 74 h

Table 1—Acute toxicological effects of pesticides.

Compound	Group	Species	Type	Effective concentration (µg/l)	Temp (°C)	Salinity (ppt)	Reference
Decamethrin	Fish	*Salmo salar*	96-h LC_{50}-S*	1.97	10	—	21
	Decapod	*Homarus americanus*	96-h LC_{50}-S	0.0014	10	30	21
Aminocarb	Fish	*Salmo salar*	96-h LC_{50}-S	8700	10	—	22
	Decapod	*Crangon septemspinosa*	96-h LC_{50}-S	200	10	—	22
EPN	Diatom	*Skeletonema costatum*	96-h EC_{50}-FT	340	20 ± 0.5	30	10
Carbophenothion	Diatom	*Skeletonema costatum*	96-h EC_{50}-FT	109	20 ± 0.5	30	10
DEF	Diatom	*Skeletonema costatum*	96-h EC_{50}-FT	366	20 ± 0.5	30	10
Ethoprop	Diatom	*Skeletonema costatum*	96-h EC_{50}-FT	8.4	20 ± 0.5	30	10
Methyl parathion	Diatom	*Skeletonema costatum*	96-h EC_{50}-FT	5.3	20 ± 0.5	30	10
Phorate	Diatom	*Skeletonema costatum*	96-h EC_{50}-FT	1.3	20 ± 0.5	30	10

*S = Static; FT = Flow Through

to 0.98 mg/l. WU *et al.* found that the new degradable DDT analog EDO (2,2-bis-(p-ethoxyphenyl)-3,3-dimethyloxetane), exhibited potent, specific actions on the sodium channel of axons of the American lobster (*Homarus americanus*).

Fate. The stability of toxaphene in estuarine sediment showed no significant degradation over 21 days.[10] These results indicated that any microbial metabolism of toxaphene in estuarine sediment that occurs does not reduce the total amount of chlorinated hydrocarbon present nor does it reduce the toxicity of toxaphene to the brine shrimp, *Artemia salina*.

Harder *et al.*[31] found that toxaphene concentrations in rainfall over a South Carolina salt marsh on occasions during and immediately following the summer use season, exceeded by several times concentrations reported to have produced damage to fish bone. The estimated aerial input of toxaphene to the 26 km^2 estuary over a 4-month period was 1.2 kg.

Wu *et al.*[36] investigated the fate of atrazine in the microsurface water of the Rhode River estuary. In the bulk surface water Atrazine concentration varied from 0.003 to 0.19 mg/l (mean = 0.04 m/l) and in microsurface water, from 0.01 to 0.59 mg/l (mean = 0.13 mg/l).

Following discovery that the James River and estuary had been contaminated with Kepone, Huggett *et al.*[33] determined that all trophic levels in the estuary contained Kepone, most being stored in fine-grained, organic-rich bed sediments and although Kepone-bound suspended material was transferred seaward, the bulk of this material accumulated in the turbidity maximum zone of the middle estuary.

Davis *et al.*[34] determined the fate and effects of Atrazine in a salt marsh ecosystem and in laboratory microecosystems. Although both the field and microecosystem tests confirmed the rapid disappearance of Atrazine from treated areas, there were significant differences and results between the two systems. Three months after Atrazine application to the marsh, the total herbicide remaining in the soil, cordgrass, snails, fiddler crabs, and horse mussels was less than 3% of that applied. Seventy days after Atrazine application to microecosystems, 50% had been removed in tidal water; 42% within the first 6 days.

Drifmiyer *et al.*[35] found that giant cordgrass *Spartina synosuroides* detritus absorbed Kepone from contaminated brackish water during decomposition. Kepone detritus concentrations increased substantially with time. Detritus accumulation provided a means of entrance of Kepone into estuarine food webs.

Duinker and Hillebrand[36] quantified organochlorines in water, seston, bottom sediment, and interstitial water in the estuarine region of the Rhine-Meuse system and found that the amounts of pentachlorobenzene and hexachlorobenzene in solution were smaller than in suspension. Dissolved concentrations were dominant for hexachlorocyclohexane isomers. During estuarine mixing, dissolved concentrations of all compounds were found to be linearly related to salinity. Wade[31] examined representative organophosphorus pesticides with respect to transport to the marine environment and rates of degradation in seawater and showed that methyl parathion and parathion were unstable in seawater, both half-lives being dependent upon the physical, chemical, and biological conditions present. Under low-nutrient growth conditions, the half-life of parathion at 27°C decreased from 90 (sterile conditions) to 41 days. Under conditions of nutrient enrichment,

parathion half-life was further reduced to 7 days. At 4°C the half-life of parathion in sterile seawater was approximately 1 year and that of methyl parathion was 188 days. At 27°C under nonsterile conditions, half-lives were 56 days and 22 days, respectively.

Spain *et al.*[38] found that prior exposure of microbial populations from different environments to pesticides can cause them to adapt to degrade the compounds more rapidly during subsequent exposures. Because biodegradation rates in laboratory systems can be affected by concentration and prior exposure, adaptation must be considered when such systems are used to predict the fate of pesticides in the environment.

Bioaccumulation. McLeese *et al.*[39] observed low uptake and high excretion rate constants of fenitrothion by soft shell clams (*Mya arenaria*) and mussels (*Mytilus edulis*). The authors concluded that organophosphates do not accumulate to any appreciable extent in aquatic organisms.

Lobsters (*Homarus americanus*) fed mussels contaminated with endrin concentrated the compound 0.41 times in the hepatopancreas and 0.001 times in the tail muscle after 2 weeks.[40] Concentrations in both tissues significantly decreased during a 4-week depuration period.

Mirex was accumulated by juvenile Atlantic salmon from both water and food.[41] After 42 days, the fish had accumulated Mirex to the level of 75 ng/g. The bioconcentration factor observed after 96 hours was 8 884. Matsuo[42] presented a theoretical treatment of the ecological magnification process in which bioaccumulation of chemicals in fish is caused by the total interactions at polar and nonpolar parts between compounds and tissues in fish and water. The author stated that based on these calculations, the "phenomenon" of bioaccumulation was easily visualized.

Residues. No significant levels of polychlorinated naphthalene (PCN) or DDT and related compounds were found in Severn Estuary sediments extracted by cylic steam distillation in hexane.[43] Wang *et al.*[44] reported that DDT levels in sediment samples of the St. Lucie estuary, Fla., ranged from 1.8 to 6.15 mg/l. No detectable DDT was found in water samples. DDT in sediment off the Turkish Coast ranged from 3 to 21 ng/g.[45]

Wade and Quinn[46] found that chlordane and DDT concentrations in sediment cores from mid-Narragansett Bay decreased with depth and reached background levels at different depths. The authors concluded that the results reflected the fact that these anthropogenic materials recorded environmental conditions of the past.

Puccett and Leoni[47] reported that HCB was identified in only 16% of 1 976 Mediterranean seawater samples (0.002 to 0.01 mg/l) analyzed and in none of the sediment samples. Brugmann and Luckas[48] detected an average of 0.14 mg/l of DDT/DDE in 36 East and North Atlantic Ocean surface water samples. Concentration variations were correlated with anthropomorphic influences. Information on biotic pesticide residues is provided in Table 2.[49-58]

RESIDUES AND RELATED TOPICS

The amount of Cd, Pb, Cu, and Zn in crustaceans and fish was measured to determine whether or not they were being concentrated in the food chain.[59,60] A transfer factor was calculated according to:

$$\text{Transfer factor} = \frac{\text{concentration in the consumer}}{\text{concentration in the prey}}$$

If the value obtained was less than one, it was assumed that there was no biomagnification. The transfer factor exceeded one only for Zn, indicating it was the only metal of the four that may be concentrated through a food chain.[59,60]

Zinc and iron were demonstrated histochemically to be cytoplasmic inclusions in the lysosomes of the mussel *Mytilus edulis*. It was suggested that this may be a method of detoxication of metalic ions in the species.[61] Experimental data suggested that metallathionine-like compounds may play a role in storage or detoxification of Cu, Zn, and Hg in diatoms.[62] There was no evidence for the binding of Cd, Zn, and Cu to metallothionein-like proteins in the barnacle *Semibalanus balanoides*.[63] The uptake of methyl-mercury was rapid in the polychaete *Glycera dibranchiata*, which could be the result of the lipid solubility and possibly the role of the sulfhydryl groups.[64]

Lead concentrations in sediments were seven times higher in sediments that received flood control run-off than a portion of the wetlands in southern California that did not.[65] The clean-up procedures for mercury-contaminated sediments in Minamata Bay were outlined.[66] Sediments were removed and used as fill material for inner parts of the bay. After completion in about 10 years, the area will be covered with high-quality soil.

The concentrations of heavy metals found in marine organisms are presented in Table 3.[63,67-107] Data on the analysis of marine organisms for the presence of metals and other elements are presented according to the element, group of organisms, and geographical region.

MICROBIOLOGY

A proceedings was published for a symposium on aquatic microbial ecology.[108] Papers included discussion of the effects of environmental stress on microorganisms; transformation of chemical pollutants; organic and heavy metal toxicity; microbiological testing under the Toxic Substance Control Act, including a discussion of the utility of "indicator organisms" such as *Aeromonas hydrophila* as indicators of environmental disturbance. Colwell[109] cautioned, however, that while microorganisms are useful indicators of alterations in the natural environment, the use of "indicator organisms" has limitations.

It has been suggested that bacteriological analyses are more suitable than chemical methods for detecting certain types of pollution.[110] An ASTM publication appeared that provided the available techniques for enumeration and activity determination of native aquatic bacteria, many of which are applicable to assessment of pollutants on estuarine and marine microbiol populations.[111]

Microorganisms as Pollutants. Information on the effects of environmental conditions on growth and survival of microorganisms currently used as indicators of pollution is presented in Table 4. Sublethal stress in *Escherichia coli* was shown to increase with salinity in various detection media.[112] Temperature and pH were found to have the greatest effect on recovery of fecal indicator bacteria.[113] Hussong *et al.*[113] found

Table 2—Pesticide residues: biota

Compound	Species	Location	Residues	Reference
Aldrin	Anchovy (*Engraulis encrasicholus*)	Central Mediterranean	0.1–0.8 ppb	49
	Striped mullet (*Mullus barbatus*)	Central Mediterranean	0.2–1.7 ppb	49
	Tuna (*Thunnus thynnus thynnus*)	Central Mediterranean	0.1–0.2 ppb	49
	Shrimp (*Nephrops norvegicus*)	Central Mediterranean	0.1–0.4 ppb	49
	Mussel (*Mytilus galloprovincialis*)	Central Mediterranean	0.4–1.7 ppb	49
BHC	Anchovy (*Engraulis encrasicholus*)	Central Mediterranean	0.2–3.4 ppb	49
	Striped mullet (*Mullus barbatus*)	Central Mediterranean	0.1–5.0 ppb	49
	Tuna (*Thunnus thynnus thynnus*)	Central Mediterranean	0.1–0.6 ppb	49
	Shrimp (*Nephrops norvegicus*)	Central Mediterranean	0.1–1.0 ppb	49
	Mussel (*Mytilus galloprovincialis*)	Central Mediterranean	0.7–3.0 ppb	49
	Anchovy (*Engraulis encrasicholus*)	Italian Ligurian Sea	0.25–1.54 ppb	50
	Shrimp (*Nephrops norvegicus*)	Italian Ligurian Sea	0.22–2.35 ppb	50
	Mussel (*Mytilus galloprovincialis*)	Italian Ligurian Sea	0.44–1.74 ppb	50
	Striped mullet (*Mullus barbatus*)	Italian Ligurian Sea	0.43–2.18 ppb	50
DDD	Cockle (*Cerastoderma edule*)	German Wadden Sea	0.6–1.0 µg/g	51
	Soft clam (*Mya aernaria*)	German Wadden Sea	3.7–6.9 ng/g	51
	Lugworm (*Arenicola marina*)	German Wadden Sea	3.1–4.5 ng/g	51
	Brown shrimp (*Crangon crangon*)	German Wadden Sea	0.6–0.8 ng/g	51
	Sole (*Solea solea*)	German Wadden Sea	7.5–13.4 ng/g	51
	Anchovy (*Engraulis encrasicholus*)	Central Mediterranean	0.8–13.4 ppb	49
	Striped mullet (*Mullus barbatus*	Central Mediterranean	0.6–15.1 ppb	49
	Tuna (*Thunnus thynnus thynnus*)	Central Mediterranean	0.3–2.9 ppb	49
	Shrimp (*Nephrops norvegicus*)	Central Mediterranean	0.1–0.4 ppb	49
	Mussel (*Mytilus galloprovincialis*)	Central Mediterranean	5.3–6.5 ppb	49
	Anchovy (*Engraulis encrasicholus*)	Italian Ligurian Sea	0.27–5 ppb	50

Table 2—(Continued)

Compound	Species	Location	Residues	Reference
	Shrimp (*Nephrops norvegicus*)	Italian Ligurian Sea	0.28–1.20 ppb	50
	Mussel (*Mytilus galloprovincialis*)	Italian Ligurian Sea	2.4–5.4 ppb	50
	Striped mullet (*Mullus barbatus*)	Italian Ligurian Sea	6.7–23.5 ppb	50
DDE	Soft clam (*Mya aernaria*)	German Wadden Sea	0.7–1.1 ng/g	51
	Lugworm (*Arenicola marina*)	German Sea Wadden Sea	0.8–1.2 ng/g	51
	Brown shrimp (*Crangon crangon*)	German Wadden Sea	0.9 ng/g	51
	Sole (*Solea solea*)	German Wadden Sea	2.8–4.6 ng/g	51
	Anchovy (*Engraulis encrasicholus*)	Central Mediterranean	4.9–27.6 ppb	49
	Striped mullet (*Mullus barbatus*)	Central Mediterranean	1.9–32.3 ppb	49
	Tuna (*Thunnus thynnus thynnus*)	Central Mediterranean	3.3–28.5 ppb	49
	Shrimp (*Nephrops norvegicus*)	Central Mediterranean	0.5–2.7 ppb	49
	Mussel (*Mytilus galloprovincialis*)	Central Mediterranean	8.2–9.5 ppb	49
	Anchovy (*Engraulis encrasicholus*)	Italian Ligurian Sea	10–15 ppb	50
	Shrimp (*Nephrops norvegicus*)	Italian Ligurian Sea	1.37–9.35 ppb	52
	Mussel (*Mytilus galloprovincialis*)	Italian Ligurian Sea	3.3–6.7 ppb	50
	Striped mullet (*Mullus barbatus*)	Italian Ligurian Sea	23.3–45.3 ppb	50
	Mullet (*Mugilauratus*)	Turkish Coastal Waters	48 ng/g	45
	Mullet (*Mullus barbatus*)	Turkish Coastal Waters	62 ng/g	45
	Mullet (*Mullus surmuletus*)	Turkish Coastal Waters	21 ng/g	45
	Goatfish (*Upeneus moluccenis*)	Turkish Coastal Waters	47 ng/g	45
	Shrimp (*Parapaneus kerathurus*)	Turkish Coastal Waters	28 ng/g	45
	Limpet (*Patella caerulea*)	Turkish Coastal Waters	2 ng/g	45
DDT	Anchovy (*Engraulis encrasicholus*)	Central Mediterranean	11.3–82.7 ppb	49
	Striped mullet (*Mullus barbatus*)	Central Mediterranean	3.9–85.5 ppb	49
	Tuna (*Thunnus thynnus thynnus*)	Central Mediterranean	6.6–51.3 ppb	49

Table 2—(Continued)

Compound	Species	Location	Residues	Reference
	Shrimp (*Nephrops Norvegicus*)	Central Mediterranean	1.2–4.5 ppb	49
	Mussel (*Mytilus galloprovincialis*)	Central Mediterranean	29.8–35.6 ppb	49
	Anchovy (*Engraulis encrasicholus*)	Italian Ligurian Sea	13–38 ppb	50
	Shrimp (*Nephrops Norvegicus*)	Italian Ligurian Sea	1.7–10.2 ppb	50
	Mussel (*Mytilus galloprovincialis*)	Italian Ligurian Sea	31.3–42.8 ppb	50
	Striped mullet (*Mullus barbatus*)	Italian Ligurian Sea	62.6–121.4 ppb	50
	Dover sole (*Microstomus pacificus*)	Southern California Coast–sewer system outfall zone	4.7–25 mg/kg	52
	Kelp bass (*Paralabrax clathratus*)	Southern California Coast–sewer system outfall zone	0.26–12 mg/kg	52
	Black perch (*Embiotoca jacksoni*)	Southern California Coast–sewer system outfall zone	2.9–25 mg/kg	52
	Sprat (*Sprattus sprattus*)	English medway Estuary	0.17–0.50 μg/g	53
	Harbour porpoise (*Phocoena phocoena*) (fetus)	North Sea	12.5 μg/g (fat)	54
	Mullet (*Mugil auratus*)	Turkish Coastal Waters	89 ng/g	45
	Mullet (*Mullus barbatus*)	Turkish Coastal Waters	130 ng/g	45
	Mullet (*Mullus surmuletus*	Turkish Coastal Waters	34 ng/g	45
	Goldband goat fish (*Upeneus moluccensis*)	Turkish Coastal Waters	74 ng/g	45
	Shrimp (*Parapaneus kerathurus*)	Turkish Coastal Waters	34 ng/g	45
	Limpet (*Patella caerulea*)	Turkish Coastal Waters	5 ng/g	45
	Deep sea fish (*Antimora rostrata*)	Atlantic Ocean East U. S. Coast	3.81–11.93 mg/kg (liver)	55
	Harbour seal (*Phoca vitulina*)	Dutch Wadden Sea	0.1–6.9 mg/kg (blubber)	56
	Mussel (*Mytilus edulis*)	Dutch Coastal Waters	0.003–0.005 mg/kg	57
	Zooplankton	Finland Coast	34–103 μg/kg (dry weight)	58
Dieldrin	Anchovy (*Engraulis encrasicholus*)	Central Mediterranean	0.1–0.8 ppb	49
	Striped mullet (*Mullus barbatus*)	Central Mediterranean	0.1–2.5 ppb	49
	Tuna (*Thunnus thynnus thynnus*	Central Mediterranean	0.1–0.4 ppb	49

Table 2—(Continued)

Compound	Species	Location	Residues	Reference
	Shrimp (*Nephrops Norvegicus*)	Central Mediterranean	0.1–0.2 ppb	49
	Mussel (*Mytilus galloprovincialis*)	Central Mediterranean	0.2–1.3 ppb	49
	Cockle (*Cerastoderma edule*)	German Wadden Sea	0.6–2.2 ng/g	51
	Soft clam (*Mya aernaria*)	German Wadden Sea	3.0–7.2 ng/g	51
	Brown shrimp (*Crangon crangon*)	German Wadden Sea	0.4–0.6 ng/g	51
	Sole (*Solea solea*)	German Wadden Sea	2.8–5.2 ng/g	51
	Sprat (*Sprattas sprattas*)	English Medway Estuary	0.03–0.14 µg/g	53
	Harbour porpoise (*Phocoean phocoena*) (fetus)	North Sea	2.5 µg/g (lipid)	54
	Deep sea fish (*Antimora rostrata*)	Atlantic Ocean East U. S. Coast	0.2 µg/kg (liver)	55
	Harbour seal (*Phoca vitalina*)	Dutch Waddan Sea	0.02–1.4 mg/kg (blubber)	52
	Mussel (*Mytilus edulis*)	Dutch Coastal Waters	0.01 mg/kg	57
Endosulfan	Cockle (*Cerastoderma edule*)	German Wadden Sea	0.2–0.3 ng/g	51
	Soft clam (*Mya aernaria*)	German Wadden Sea	0.5–0.8 ng/g	51
	Lugworm (*Arenicola marina*)	German Wadden Sea	0.3–0.5 ng/g	51
	Brown shrimp (*Crangon crangon*)	German Wadden sea	0.3–0.4 ng/g	51
	Sole (*Solea solea*)	German Wadden Sea	0.9–1.7 ng/g	51
Endrin	Mussel (*Mytilus edulis*)	Dutch Coastal Waters	0.03–0.13 mg/kg	57
HCB	Mussel (*Mytilus edulis*)	Dutch Coastal Waters	0.001–0.016 mg/kg	57
Hexachlorobenzene	Harbour porpoise (*Phocoena phocoena*)	North Sea	0.1 µg/g (lipid)	54
Mirex	Harbour seal (*Phoca vitalina*)	Dutch Wadden Sea	0.1–1.3 mg/kg (blubber)	56
TDE	Mussel (*Mytilus edulis*)	Dutch Coastal Waters	0.003–0.008 mg/kg	57

that the number of false-positives for total coliforms detected in the confirmation test of MPN evaluation increased significantly when the water temperature dropped below 10°C and suggested using the completed test under these conditions. In natural aquatic systems, survival of enteric bacteria was shown to be controlled by such mechanisms as protozoan predation[115] and attachment to sediments.[116] In the latter, organic content appeared to be a determining factor in survival.

Microbiological monitoring of nearshore waters has demonstrated the need for improved wastewater treatment in several coastal cities.[117,118] Likewise, detection of coprostanol, a microbiological degradation product of cholesterol excreted in mammalian feces, demonstrated serious wastewater pollution in one region of the harbor of Mazatlan, Mexico.[119]

Bacterial disease. Microbiological monitoring of commercial seafoods continued to reveal the presence of *Vibrio parahaemolyticus*.[120] Larval oysters infected with several marine *Vibrio* spp. demonstrated decreased growth and/or high mortality.[121] One bacterial isolate attached to the larval shell and destroyed mantle tissue. The extent of severity of fin rot in Atlantic salmon was shown to be a function of temperature.[122] The etiological agent appeared to be either a *Flexibacter* sp.

or *Aeromonas salmonicida* depending on the hatchery. A new plate-count agar was developed for enumeration of aerobic, heterotrophic bacteria in estuarine waters, sediments and shellfish.[123]

Viruses. Progress continues to be made in the evaluation of viral contamination in estuarine and coastal waters. Metcalf *et al.*[124] proposed an improved method and test strategy for recovery of enteric viruses from shellfish. Kapuscinski and Mitchell[125] reviewed processes controlling virus inactivation in coastal waters. They concluded that biological and chemical inactivation were most important. Virus adsorption to sediment greatly increased survival time.[126] This survival mechanism seemed to only function at high (37°C) temperatures for poliovirus.[127] It was demonstrated that enterovirus uptake by the soft-shelled clam, *Mya arenaria*, was enhanced when viruses were associated with particles.[128] Virus elimination occurred within a 24 to 48-h depuration. Correspondingly, LaBelle, *et al.*[129] found greater numbers of viruses in sediments than in overlying water. After examining a number of parameters, the only significant association was a positive correlation between viruses and total fecal coliforms in the sediment. Comparison between waters open and closed to shell-fishing revealed very little difference in viral quality.[130] While viruses were frequently isolated from treated wastewater effluents, very few were ever detected in receiving surface waters in Oyster Bay, N. Y. While the presence of virucidal agents in estuarine and marine waters has been recognized for some time, their identity has been elusive. Heavy metals[125] and microbial agents[131] have been suggested as possible sources of virucidal activity. Survival of simian rotavirus, which exhibited stability in natural waters similar to enteroviruses, was affected by a non-seasonal change in water quality.[132]

Pollutant-microbe interactions. The surface microlayer of a salt marsh in San Francisco Bay was found to contain elevated concentrations of particulates, microorganisms, and lead.[133] The investigators suggested that microlayer particulate surfaces may act as a site for uptake of lead into marine biomass. Aerosols of marine origin formed from surface microlayers may therefore present a health hazard to populations inhabiting small islands.[134]

High levels of Cu, Pb, and Zn caused local inhibition of bacterial heterotrophic processes and loss of viability in a slough of an industrialized estuary receiving metal finishing wastes.[135] Bacteria and metals did not generally influence each other in other areas of the estuary. In laboratory experiments, viable counts of various physiological groups of bacteria cultured in lead-contaminated artificial seawater were not significantly different from those cultured without lead enrichment. Lead tolerance was lost, however, when bacteria were cultured in a medium without lead.[136]

Microorganisms isolated from sediments of the New York Bight apex that were resistant to cadmium were also found to exhibit drug-resistance.[137] Another study further demonstrated that drug-resistance could be transferred between strains of *E. coli* in sediments of the New York Bight contaminated with wastewater sludge.[138] The capacity of sediment microbes to modify the state in which metals exist was demonstrated.[139]

Several reports accessed the relationship between concentrations of oil-degrading bacteria and the degree of oil pollution in marine waters.[140–142] Petroleum products and brine discharge from off-shore drilling platforms did not seem to be toxic to marine bacteria.[141] Likewise, the utilization of glucose by marine and estuarine bacterial populations was usually not affected by the presence of crude oil.[143]

Microbial degradation of selected hydrocarbons was assessed in relation to sediment conditions.[143–145] Oxygen concentration, oxidation-reduction potential, and pH were found to be important factors. Enclosed ecosystems were used to examine various aspects of hydrocarbon degradation in seawater.[146,147] In one of the studies, 230 days were required to oxidize 29 percent of the initial quantity of benzanthracene present to CO_2.[147] Understanding of the pathways of mono- and bicyclic aromatic hydrocarbon degradation were described for an *Arthrobacter* sp. isolated from seawater.[148]

As lists of toxic substances expand, new studies demonstrate the capacity of estuarine and marine microbial populations to tolerate and transform previously untested compounds.[149,150] A topic that has received increased attention is the chemical consequences of microbially mediated degradation of the numerous organic pollutants which find their way into estuarine and marine systems.[151,152] The impact of pollution on the utilization and mineralization of important inorganic nutrients such as N, S, and P by microorganisms has also been examined in coastal waters.[153,154] A list of references relating to microbe-pollutant interactions is presented in Table 5.

Several studies have applied techniques combining both microbiological and chemical analyses for impact studies. Alexander and Boethling[155] developed a sensitive method for measuring biodegradation of organic chemicals based on the formation of $^{14}CO_2$ from ^{14}C-labelled compounds. Using this technique they found that biodegradation rates at low substrate concentrations were well below those predicted by Michaelis-Menten kinetic relationships. Kennicut[156] proposed measuring ATP concentration, a measure of living biomass, as a fast and reliable indicator of toxicity.

Surveys. Word[162] sampled soft-bottom benthic communities in deeper water of the southern California Bight, which enabled extension of the Infaunal Trophic Index (ITI) to depths ranging from 20 to 800 m. Natural communities at deep sites distant from major sources of organic pollutants were dominated by suspension-feeding taxa and had ITI values greater than 83. Additional analyses of deepwater stations resulted in a list of 53 invertebrate taxa (compared with 46 taxa used previously) which could be separated into 4 infaunal trophic index feeding groups.[163] Criteria for trophic group assignments were presented in addition to specific examples and descriptions of feeding behavior. Evaluation of the effects of replication of the ITI indicated that a single 0.1 m^2 sample was adequate to characterize benthic communities within a range of 5 ITI units at the 95% confidence level.[164] Bellan[165] proposed an Annelid Index of Pollution for rocky substrate polychaete communities, based on the ratio of the sums of percent occurrences of pollution indicator species (for example, *Platynereis dumerili, Dorvillea rudolphi* or *Theostoma oerstedi*) to the sums of percent occurrences of "pure water" species (for example, *Syllis* spp. or *Amphiglena mediterranea*). In an extensive field study on the French Mediterranean coast, the index was well correlated with the Shannon-Weaver diversity index and allowed for discrimination of samples from heavily

Table 3—Chemical residues reported from marine organisms.

Element	Type of organism	Concentration (µg/g Dry weight), Locality & Reference
Aluminum	Pelecypods	60.2–518, Calif.[66]
Arsenic	Green Algae	15.3–113.6, U. K.[68]
	Brown Algae	35.5–189.3, U. K.[68]
	Brown Algae	21–32, Canadian Arctic[69]
	Anthozoa	53.3–59, U. K.[68]
	Pelecypods	16.8–17.2, U. K.[68]
	Gastropods	35.3–97.7, U. K.[68]
	Gastropods	4.6–27*, U. K.[70]
	Decapods	19.9–34.5, U. K.[68]
	Echinoids	2.7–16, Canadian Arctic[69]
Cadmium	Green Algae	0.1–0.3, U. K.[68]
	Brown Algae	0.2–3.8, U. K.[68]
	Brown Algae	0–36**, Norway[71]
	Brown Algae	0.7–1.8, Canadian Arctic[69]
	Zooplankton	0.3–10.9, French Mediterranean[70]
	Anthozoa	0.3–0.4, U. K.[68]
	Polychaetes	0.1, U. K.[73]
	Pelecypods	2.49–4.82, Calif.[66]
	Pelecypods	14.1–292, R. I.[74]
	Pelecypods	0.005–0.8, Quebec[75]
	Pelecypods	1.12–3.2, Quebec[75]
	Pelecypods	0.25–0.59*, U. K.[76]
	Pelecypods	0.8–3.2, U. K.[68]
	Pelecypods	2.5–541**, Germany[78]
	Pelecypods	1.1–34.1, Germany[77]
	Pelecypods	0.2–365, South Africa[79]
	Pelecypods	0.76–318.6, Australia[80]
	Gastropods	13.2–17.7, U. K.[68]
	Barnacles	0–60**, U. K.[84]
	Decapods	0.1–148*, U. K.[81]
	Echinoids	0.6–1.8, Canadian Arctic[69]
	Fish	0.001–12.8, Germany[82]
	Birds	0–480, U. K.[68]
	Birds	0–336, U. K.[84]
	Mammals	0.1–209, Australia[85]
Calcium	Protozoans	7,260; Atlantic Ocean[86]
	Polychaetes	0–3,480; U. K.[87]
	Polychaetes	572–710, U. K.[88]
	Polychaetes	531–1,130; U. K.[73]
	Barnacles	420–2,870**; U. K.[89]
	Amphipods	73,700–74,400; U. K.[90]
Chromium	Pelecypods	2.13–3.52, Calif.[67]
Cobalt	Pelecypods	0.022–0.175, Wash.[95]
	Mammals	0.5–72, Australia[85]
Copper	Green Algae	62.3–69.8, U. K.[68]
	Green Algae	1.24–3.2, Pacific Islands[91]
	Brown Algae	5.75–8.8, Japan[92]
	Brown Algae	2.1–4.7, Canadian Arctic[69]
	Brown Algae	62.4–228.3, U. K.[68]
	Zooplankton	5.9–129.1, French Mediterranean[72]
	Protozoans	640, Atlantic Ocean[86]
	Anthozoans	7.6–7.9, U. K.[68]
	Polychaetes	32–130,400; U. K.[87]
	Polychaetes	2–15, U. K.[88]
	Polychaetes	4–80, U. K.[73]
	Pelecypods	0.96–1.81*, U. K.[76]
	Pelecypods	7.6–2,220; U. K.[68]
	Pelecypods	1.49–10.4, South Africa[79]
	Pelecypods	0.03–5, Quebec[75]
	Pelecypods	11–325, British Columbia[93]
	Pelecypods	4.3–111.1**, Roesijadi[94]
	Pelecypods	5.63–8.06, Calif.[67]
	Gastropods	14.6–276.1, U. K.[68]
	Barnacles	232–3,232**, U. K.[84]
	Amphipods	77–259, U. K.[90]
	Decapods	4–206*, U. K.[81]
	Echinoids	2.3–9, Canadian Arctic[69]
	Mammals	9.1–608, Australia[85]
Iron	Green Algae	15.15–72.66, Pacific Islands[91]
	Brown Algae	34.3–99.3, Japan[92]
	Brown Algae	250–920, Canadian Arctic[69]
	Protozoans	1,460; Atlantic Ocean[86]
	Polychaetes	34–2,430; U. K.[87]
	Polychaetes	95–1,089; U. K.[88]
	Polychaetes	44–1,180; U. K.[73]
	Pelecypods	246–661, Calif.[67]
	Pelecypods	10–500, Quebec[75]
	Pelecypods	4.1–134, South Africa[79]
	Barnacles	1.6–120**, U. K.[89]
	Amphipods	325–494, U. K.[90]
	Echinoids	185–2,610; Canadian Arctic[69]
	Mammals	778–82,367; Australia[85]
Lead	Green Algae	0–3.69, Pacific Islands[91]
	Brown Algae	0.5–1.4, Canadian Arctic[69]
	Protozoans	550, Atlantic Ocean[86]
	Polychaetes	8.7, U. K.[73]
	Pelecypods	2.03–2.9, Calif.[67]
	Pelecypods	0–470, British Columbia[93]
	Pelecypods	0.5–4.1, Quebec[75]
	Pelecypods	0.43–2.6*, U. K.[76]
	Pelecypods	0.07–4.16, South Africa[79]
	Amphipods	60, U. K.[90]
	Fish	0.1–0.5*, Italy[97]
Magnesium	Polychaetes	890–5,270; U. K.[87]
	Polychaetes	464–592, U. K.[88]
	Polychaetes	616–1,080; U. K.[73]
	Amphipods	11,600; U. K.[90]
Manganese	Green Algae	4.63–9.45, Pacific Islands[91]
	Brown Algae	3.01–5.06, Japan[92]
	Polychaetes	1–17.2, U. K.[87]
	Polychaetes	380–1,079; U. K.[88]
	Polychaetes	510–952, U. K.[73]
	Pelecypods	1.9–33.9, South Africa[79]
	Pelecypods	0.1–10, Quebec[75]
	Barnacles	20–70**, U. K.[89]
	Amphipods	32–33, U. K.[90]
Mercury	Brown Algae	0.02–1.63**, Norway[71]
	Phanerogam	2–142.6, French Mediterranean[72]
	Zooplankton	0.02–0.12, Tropical Pacific Ocean[98]
	Pelecypods	0.03–0.46*, U. K.[76]
	Pelecypods	0.009–0.016, India[99]
	Cephalopods	0.006–0.04, India[99]
	Decapods	0.005–0.055, India[99]
	Decapods	10–40*, Turkey[100]

Table 3—(Continued)

Element	Type of organism	Concentration (μg/g Dry weight), Locality & Reference
	Fish	2–130*, Turkey[100]
	Fish	0.01–5.4, Italy[97]
	Fish	0.1–1,021**, Spain[101]
	Fish	0.001–0.12, India[99]
	Fish	0.09–0.21*, India[102]
	Fish	0.24–3.27, Malaysia[103]
	Fish	0.1–82, Australia[104]
	Fish	0.03–0.75, Australia[105]
	Birds	0–61.5, U. K.[83]
	Birds	0–30.4, U. K.[84]
Nickel	Brown Algae	2.7–3.38, Japan[92]
	Pelecypods	1.82–3.16, Calif.[67]
	Pelecypods	0.05–2, Quebec[75]
	Pelecypods	0.4–1.86, South Africa[79]
Potassium	Polychaetes	129–151, U. K.[87]
	Polychaetes	99–385, U. K.[88]
	Polychaetes	270–960, U. K.[73]
Rubidium	Protozoans	1,680; Atlantic Ocean
Selenium	Birds	0–89.1, U. K.[84]
Silver	Polychaetes	0.1, U. K.[86]
	Mammals	0.2–39, Australia[85]
Sodium	Polychaetes	644–1,650; U. K.[87]
	Polychaetes	549–1,290; U. K.[88]
	Polychaetes	1,150–2,690; U. K.[73]
Zinc	Green Algae	182–206.5, U. K.[66]
	Green Algae	3.73–10.88, Pacific Islands[91]
	Brown Algae	22–38, Japan[92]
	Brown Algae	513–1,441; U. K.[66]
	Brown Algae	50–3,390**, Norway[71]
	Brown Algae	22–170, Canadian Arctic[69]
	Zooplankton	52–742, French Mediterranean[72]
	Protozoans	620, Atlantic Ocean[96]
	Anthozoans	286–294.5, U. K.[66]
	Polychaetes	210–4,500; U. K.[87]
	Polychaetes	88–34,940; U. K.[66]
	Polychaetes	10,130–18,360; U. K.[73]
	Pelecypods	17.4–43.6*, U. K.[76]
	Pelecypods	1,602–7,207; U. K.[66]
	Pelecypods	16–118, South Africa[79]
	Pelecypods	1–100, Quebec[75]
	Pelecypods	155–673, British Columbia[93]
	Pelecypods	0.24–3.4, Washington[95]
	Pelecypods	182–226, Calif.[67]
	Gastropods	240.2–246.2, U. K.[66]
	Barnacles	76.5–8.470**, U. K.[80]
	Barnacles	27,837–113,250**, U. K.[64]
	Amphipods	104–109, U. K.[90]
	Decapods	188.8–211.6, U. K.[66]
	Decapods	12–95*, U. K.[81]
	Decapods	8–105.4, U. K.[106]
	Echinoids	22–184, Canadian Arctic[69]
	Fish	99.6–129, U. K.[106]
	Fish	39.6–620.5**, Spain[107]
	Birds	8–1,123; U. K.[83]
	Mammals	219–4,183; Australia[85]

* wet weight
** laboratory study

Table 4—Microbial contaminants in aquatic habitats.

Organism(a)	Habitat	Reference
Coliforms	coastal waters, embayments	108, 114
Fecal coliforms & fecal streptococci	coastal waters, embayments,	108
	brackish water,	113
	sediments	129
Escherichia coli	coastal dump site sediment,	138
	saltwater media,	112
	estuary,	114
	coastal waters	117, 118
Enterobacter spp.	coastal waters, embayments	108
	sediments	116
Aeromonas hydrophila	coastal waters	108
Vibrio spp.	estuaries	108
	oysters	121
	fish	120
Flexibacter, Aeromonas salmonicida	Atlantic salmon	122
Simian rotavirus	estuary	132
Enterovirus	shellfish	124, 128
	estuarine sediments	126, 127, 129
	subtropical ocean water	130, 131

polluted environments (AIP > 1), although the assemblages were qualitatively similar to clean water sites. Amphipod communities in rocky habitats of the French Mediterranean coast also displayed species distribution patterns associated with degree of pollutant stress.[166] No characteristic species group was associated with polluted sites; however, abundances of

Table 5—Microbial—pollutant interactions.

Compound	Conditions	Reference
Heavy metals (general)	various physical & chemical	108
	industrialized estuary	135
Mercury	Mediterranean sediments	139
Lead	salt marsh surface microlayer	133
Cadmium	New York Bight sediments	137
Arsenic	transformations under laboratory conditions	150
Kepone	Chesapeake Bay degradation under laboratory conditions	157, 158
Organophosphorus pesticides	wastewater microorganisms	159
Methyl parathion, p-nitrophenol, p-aminophenol	salt marsh, estuary	160, 161
Hydrocarbons, petroleum products	sediments	145, 146
	coastal waters	142
Crude oil	seawater lagoons	146
	estuarine, marine	143
	coastal waters laboratory study	140, 141, 148
Benzanthracene	seawater enclosures	147
Phenolics	general	149
Methylamines, ethylamines	Aquatic—general	155

Caprella acutifrons, Podocerus variegatus and *Jassa falcata* were much higher near pollutant sources. Alternatively, the relative abundances of *Hyale* spp., *Caprella liparotensis* and *Elasmopus pocillimanus* declined considerably as the degree of pollutant stress increased. The introduced enchytraeid oligochaete, *Lumbricillus lineatus*, was dominant within 1.5 km of a pulp mill waste outfall in British Columbia.[167] The percentage of total adult enchytraeids represented by *L. lineatus* was proposed as an index of pollutant impact, because a diverse enchytraeid assemblage without *L. lineatus* was observed at undisturbed intertidal sites.

At a wastewater outfall in lower Chesapeake Bay, densities of living foraminifers increased with distance from the outfall.[168] A station in the immediate outfall vicinity had a significantly different foraminiferal population than sites 100 to 300 m from the outfall. The outfall station had the lowest number of species, but species diversity (H') and equitability (E) were higher because of the lack of dominance by one or two species. The soft-bottom benthic community near a wastewater outfall in Victoria, Australia, was composed of polychaetes and crustaceans and was dominated by suspension and deposit feeders.[169] Species diversity (H'), evenness (J), and number of species at sites 500 to 1 900 m from the outfall were comparable to those at nearby sites with similar sediment characteristics although nearshore organic enrichment may cause some stimulation of spionid polychaetes and the ghost shrimp, *Callinassa ceramica*. Effects of phosphate-rich water from Port Phillip Bay were detectable for approximately 28 km offshore of the Victoria Coast.[170] Benthic community parameters in the Belfast Lough varied with distance from Belfast, which was the primary source of municipal and industrial wastes into this estuarine system.[171] The gross pollution effects, as evidenced by high oligochaete densities and low numbers of amphipods, were confined primarily to dock areas near Belfast. Andrews and Rickard[172] reported a secondary phase of biological recovery of the inner Thames estuary following improved effluent quality at a large municipal waste treatment facility. The initial recolonization after 1964 followed establishment of aerobic conditions during the summer. The second phase (1974–80) was characterized by increased numbers of species of invertebrates and fishes with reproduction of both groups occurring in the estuary. A study of phytoplankton in Kaneohe Bay, Hawaii, which receives wastewater discharge, indicated that nitrogen was a limiting factor in all areas of the bay.[173] However, phosphorus was simultaneously limiting in the more oligotrophic areas farthest from wastewater outfalls. Because of high uptake of the inorganic nutrients in the enriched embayment, particulate concentrations such as chlorophyll *a* and ATP were suggested as more sensitive indices of eutrophication. Hannah and Boney[174] compared the contribution of nonoplankton (<20 μm) to the total phytoplankton biomass and carbon fixation in two neighboring sea lochs receiving different wastewater loads. In the more heavily loaded Gareloch, the contribution of nanoplankton to both parameters increased markedly following thermal stratification, while the relative amount of nanoplankton remained constant in Loch Long.

Effluent from a titanium-dioxide factory resulted in decreases in intertidal and benthic fauna near the discharge; however, gross ecological changes have apparently not occurred.[175] Large-scale discharge of industrial and municipal effluents in combination with inadequate tidal flushing apparently caused nutrient enrichment of Visakhapatnam Harbor, India.[176] Phytoplankton blooms were dominated by *Skeletonema costatum*. Analysis of intertidal macroalgal assemblages on the southern California coast by reciprocal averaging ordination indicated a distinct latitudinal gradient south of Point Conception.[177] However, the gradient was not strictly latitudinal and factors such as water temperature, sand movement, human disturbance, and wave action were suggested as being responsible for deviations near the Ventura-Port Hueneme and north Los Angeles areas. Frequent anoxia of bottom waters in areas of high phytoplankton productivity in a Danish fjord occasionally resulted in mass mortalities of benthic organisms and also affected benthic community composition.[178]

Past studies have indicated the common occurrence of floating plastic material in coastal areas throughout the world. Based on neuston samples collected from the South Atlantic Ocean, Morris[179] reported that polyethylene and polypropylene pellets, together with tar balls, were the most common floating contaminants (1 000–2 000 particles/km^2). A count in the Eastern Mediterranean[180] indicated that the average density of large particles (>1.5 cm) was approximately 2 000/km^2 and plastic was the most commonly observed material.

Effects of dredging and harbor construction were studied in several habitats. Following dredging and blasting in a Central Indian Ocean atoll lagoon, species richness of coral fauna was high and was comparable to nearby unaltered lagoons.[181] Swartz *et al.*[182] reported that richness and density of macrobenthic species declined 52% and 20%, respectively, following initial dredging at Yaquina Bay, Oregon. During early stages of recovery, a dominance of opportunistic species was observed, associated with adult immigration of the amphipod *Anisogammarus pugettensis* and larval recruitment of the polychaete *Armandia brevis*. About 1 year was required for full recovery of benthic communities to pre-dredging conditions. Potential effects of large-scale sand dredging in the North Sea were assessed by de Groot.[183] Effects were predicted for plankton, bottom fauna, and fish with full recovery requiring about 3 years.

COMPLEX EFFLUENTS AND ENVIRONMENTS

Papers focused on toxicity of contaminants to marine and estuarine animals are summarized in Table 6. W. H. Thomas and coworkers[184–186] studied the effects of a mixture of 11 heavy metals (Cu, Zn, Ni, Cr, Pb, Cd, Hg, As, Ge, Sb, and Se) on phytoplankton growth in a natural assemblage of diatoms and dinoflagellates from Saanich Inlet, B. C., Canada, and in laboratory cultures of the diatom *Thalassiosira aestivalis*. Morphological effects of the individual metals on the laboratory cultures of *T. aestivalis* were compared and found to be divided into three groups according to toxic effects: Cu, Zn, Ge; Hg, Cd, Pb; and Cr, Ni, Se, Sb, and As. Similarity of morphological effects caused by Cu, Zn, and Ge suggested similar biochemical pathways of Si metabolism were affected. The toxicity of Cu, Zn, Cd, and Pb to the diatom *Ditylum brightwelli* decreased as EDTA concentration was increased, whereas increased levels of EDTA had no effect on Hg, Ag and Ti toxicity.[187] Kaiser[188] predicted the levels of metals that

Table 6—The effects of trace contaminants on marine organisms.

Toxicant(s)	Group	Species	Observed effects	Effective concentration (mg/l)*	Reference
As, Cd, Cu, Cr, Ge, Hg, Ni, Pb, Sb, Se, Zn	Diatoms	*Thalassiosira aestivalis*	Changes in morphology, inhibition of natural assemblages, inhibition of growth		184–186
Ag, Cd, Cu, Pb, Zn, Hg, Thallium	Diatom	*Ditylum brightwelli*	Decreased metal (Cd, Cu, Pb, Zn) toxicity with increased EDTA concentration		187
Cd, Cu, Pb	Herring eggs and larvae	*Clupea harvengus*	Changes in embryonic survival, viable hatch and uptake of metals		189
Cd, Cu, Zn	Shrimp	*Crangon crangon*	Differences in mortality rates, sensitivities of different moulting stages and feeding rates		190
Cd	Shrimp	*Palaemonetes pugio*	Incipient LC_{50}	0.2–0.3	205
	Hermit crab	*Pagurus longicarpus*	Incipient LC_{50}	0.07	
	Bay Scallop	*Argopecten irrodians*	Incipient LC_{50} 42 day EC_{50}: growth rate	0.53 0.078	
Cu	Phytoplankton		Particulate nitrogen settling in the water. Column was reduced due to a decline in zooplankton grazing in presence of copper.		191
Cu	Algae	*Pelvetia canaliculata*	Reductions in growth–10 das.	0.012	206
		Fucus spiralis	Reductions in growth–10 das.	0.012	
		Fucus vesiculsus	Reductions in growth–10 das.	0.05	
		Fucus serratus	Reductions in growth–10 das.	0.025	
Cu	Clams	*Protothaca staminea*	LT_{50}—17 days LT_{50}—12 days	0.039 0.082	192
Cu	Barnacle navplii	*Balanus improvisus*	24-h LC_{50} at different salinities:		187
			15‰		0.088
			30‰	>0.200	
			Reduced photokinesis	0.1	
Cu	Shrimp	*Pandalus danae*	Lethal response	0.02 (>0.007 labile copper)	207
			Moulting delay	(<0.001 labile copper)	
Hg	Diatom	*Skeletonema costatum*	Increases in mercury sensitivity as ammonium becomes more limiting to diatom growth		183
Hg		*Corophium volutator*	Avoidance of mercury-treated sediments	0.001	194
Zn	Phytoplankton		Metals were detoxified by brown algal polyphenols, which allowed for more normal phytoplankton growth		201
Barita	Meiofauna		Changes in sediment population due to settling of barite		202
Fe (iron ore dust)	Algal oospheres	*Fucus serratus*	Dust laden oospheres are less effective attractants of spermatozoids		203
Phenol	Minnow	*Phoxinus phoxinus*	Lethal threshold concentration	10	208
	Isopod	*Mesidotea entomon*	Lethal threshold concentration	85–100	

Table 6—(Continued)

Toxicant(s)	Group	Species	Observed effects	Effective conventration (mg/l)*	Reference
	Amphipod	*Gammarus duebeni*	Lethal threshold concentration	32–41	
Tolvene	Medaka	*Oryzias latipes*	96-h LC_{50}	54	209
Biocide (alimicide C-30)	Mussel	*Mytilus galloprovicialis*	96-h LC_{50}	0.07	210
	Mysids	*Leptomysis mediterranea*	96-h LC_{50}	0.11	
Chlorine	Crab	*Cancer productus*	Fourfold increase in ammonia excretion in 96h	0.68 TRO	196
Chlorinated sewage waste	Spot	*Leiostomas xanthurus*	96-h LC_{50}	0.23	195
Bromochlorinated sewage waste	Spot	*Leiostomas xanthurus*	96-h LC_{50}	0.25	195
Domestic sewage effluents	Macro-algae		Responses to long-term laboratory exposure. Specific differences in tolerance		211
Sewage effluents	Sea urchin gametes and larvae	*Strongylocentrotus purpuratus*	Decreases in fertilization and development percentages with increases in sewage concentrations		197
Secondary treated textile waste		*Skeletonema costatum*	Lethal and sublethal effects		199
		Paleomonetes pugio	Lethal and sublethal effects		
		Cyprinodon variegatus	Lethal and sublethal effects		

* Effective concentrations are only included where feasible

would cause toxic effects in aquatic environments based on an understanding of electron configuration.

Eggs of Baltic herring were incubated in seawater containing mixtures of Cd, Cu, and Pb. Negative effects of the metals on embryonic survival and viable hatch were additive in the case of Cu and Cd, but Pb did not exert detrimental effects.[189] The crustacean, *Crangon crangon* was exposed to three metals, Cd, Cu, and Zn, at 0.35, 1.6 and 14.4 mg/l, respectively. Moult-stage-dependent mortality was not observed for Cd, but in Cu and Zn the post-moult stages were more susceptible than the pre-moult or inter-moult stages.[190]

Uptake and regeneration of nitrogen, and the effects of low concentrations of Cu, were investigated in Controlled Experimental Ecosystems (CEEs). In copper-treated CEEs, the amount of particulate nitrogen settling through the water column was substantially less than the controls because of a decline in zooplankton grazing in the presence of copper.[191] Roesijadi[192] exposed the clam *Protothaca staminea* to Cu concentrations of 0.007 to 0.082 mg/l for 30 days and found these concentrations were stressful with the gills being the primary organs of concentration. The regulation of Na and K by the clam was disrupted at 0.039 mg/l, and a sublethal cytotoxic response was evident at lower concentrations.

The effect of sublethal additions of mercuric chloride on the diatom *Skeletonema costatum* grown in NH_4-limited chemostats and batch cultures was assessed by Cloutier-Mantha and Harrison.[193] Ammonium-limited diatom cultures were substantially more sensitive to mercuric chlorides than non-ammonium-limited cultures, implying that Hg pollution may seriously decrease the ability of a species to utilize a limiting nutrient during periods of seasonal limitation. In another Hg experiment, burrowing behavior of *Corophium volutator* was recorded in sediments treated with seawater solutions of 0.001 to 1 000 mg/l $HgCl_2$. *C. volutator* avoided treated sediments (0.001 to 0.1 mg/l) even through bacterial food was still available in the sediments.[194]

In experiments with complex effluents and seawater chlorination, Roberts[195] found the 96-h LC50 for the spot *Leiostomas xanthurus* to be 0.23 and 0.25 mg/l for chlorinated and bromochlorinated treated wastewater, respectively. Crabs, *Cancer productus*, exposed to chlorinated seawater showed alterations in haemolymph Na and Mg concentrations because of changes in bladder wall activity.[196]

Oshida and Goochey[197] tested sea urchin fertilization and larval development in dilute seawater solutions of three southern California effluents and found all to be toxic at about 1 to 20% effluent in seawater. Bougis *et al.*[198] have proposed using sea urchin larvae to assess seawater quality.

Effects of secondary waste effluents from textile manufacturing plants were determined on *Skeletonema costatum, Palaemonetes pugio* and *Cyprinodon variegatus*. The wastes had minimal effects on *P. pugio* and *C. variegatus*, but was in-

hibitory to *S. costatum* at concentrations over 50%.[199] The mortality, growth, and fecundity of mussel, *Mytilus edulis*, and barnacle, *Balanus glandula*, populations near a kraft pulp mill outfall were determined. The adults or both species were tolerant to the effluent but growth rates were retarded.[200]

Ragar *et al.*[201] examined growth responses of several species of phytoplankton, cultured with and without metal stress, to supplements of brown algal polyphenols. The polyphenols significantly increased the ability of the phytoplankton to grow by detoxifying some of the zinc. Polyphenols from brown algae may contribute to the natural chelating capacity of inshore seawater.

Cantelmo *et al.*[202] studied the effects of barite, a major constituent of drilling muds used in marine oil drilling operations, on meiofauna in an experimental flow-through system. Boney[203] found that iron ore dust decreased the attractant ability of algae, *Fucus serratus*, oospheres for spermatozoids. Chungue *et al.*[204] examined the toxic effects of various algae. Fifteen strains of algae were cultured in an artificial medium and most were found to be non-toxic to mice.

OIL POLLUTION

Impacts of the petroleum industry on marine ecosystems was reviewed from several viewpoints including world input to the oceans[212] and French coast[213], biotransformation[214], oil spill technology[215–221], hydrocarbon characterization[222], toxicity[223], blowouts[224], and liability.[225]

Analytical Methods. A series of petroleum hydrocarbons of geochemical significance, for example C_{20}-C_{40} acyclic isoprenoids and C_{27+} steranes and triterpanes, were proposed as passive tags for the characterization of oils in the marine environment.[226] Anderson *et al.*[227] documented procedures which gave marked improvement in the identification of spilled light fuel oils by infrared spectroscopy. Methyl-to-methylene carbon and methylene-to-methine ratios were proposed as main indices of oil biodegradation as studied by ^{13}C nuclear magnetic resonance spectrometry.[228] Intralaboratory precision in gas chromatographic determination of hydrocarbons in mussels was determined to be ±40%.[229] Among commercially available enzyme preparations, glucose-6-phosphate dehydrogenaoe served as the best indicator of refinery effluent toxicity.[230]

Papers concerned specifically with polycyclic aromatic hydrocarbon analysis were reviewed.[231–232] Two papers described the analysis of polynuclear aromatic hydrocarbons from sediments.[233–234] A technique for the analysis of bioassay waters for benzo(a)pyrene in the 0.2 μg/l to 200 μg/l range was developed.[235] The application of high pressure liquid chromatography for the analysis of hydroxylated benzo(a)pyrene metabolites in marine fish was presented.[236] The analysis of polymethyl-naphthalene and -phenonthrene in marine tissues by mass fragmentography was detailed.[237]

Residues—Water and Sediment. Seawater samples from the Sargasson Sea, western equatorial Atlantic, and the upwelling region off Peru contained pentadecane, C_2-alkylated benzenes, and other hydrocarbons at levels between 2 to 60 ng/kg.[238–239] Levels of non-volatile hydrocarbons in seawater off Australia varied from less than 0.1 μg/l to over 22 μg/l with highest concentrations found close to shore-based imputs such as oil refineries.[240] Open-ocean, non-polluted Gulf of Mexico surface waters were marked by C_6-C_{14} hydrocarbon concentrations around 60 ng/l, while those from the Caribbean contained about 30 ng/l.[241] Petroleum hydrocarbons averaged from 6.9 to 25.8 μg/l at 1-m depths from the Alboran Sea and Libyan coastal regions of the Mediterranean.[242] Similar levels at depths down to 10 m were reported from northern Indian Ocean waters.[243] A possible direct contribution of phytoplankton to hydrocarbons in seawater was inferred from Mediterranean Sea data by Goutz and Saliot.[244] The presence of pelagic tar was documented.[245–247] Results of a 4-month study on the weathering of an Arabian light crude confirmed field observations of relative increases in asphaltenes and resins during the degradative time period.[248]

Age-dated sediment cores from opposite shores of the U. S. confirmed a decrease in petroleum and synthetic hydrocarbon levels as deeper sediments were analyzed.[249] Hydrocarbons in sediments were analyzed from numerous parts of the world at various depths.[250–252]

Residues—Biota. Alkane profiles for 48 bottom-dwelling organisms from an oilfield in the Gulf of Mexico revealed that only a small proportion of the mollusks studied contained petroleum hydrocarbons.[253] Benzo(a)pyrene concentrations in oysters from a Texas bay ranged from 0.07 to 0.14 ng/g or several orders of magnitude lower than those reported in 1972.[254]

Dispersants. Chemical demulsification of residual oils in seawater was modeled using aerosol or aerosol+polyethylene oxide combinations.[223] The performance of a novel "anti-mousse" additive was documented following laboratory and semi-technical scale testing.[256–257] Dispersion and weathering of treated and untreated research spills were studied in the waters of New Jersey.[258] The toxicity of petroleum solidifying agents to a marine diatom and fish was on the order of several g/l.[259] An ubiquitous interstitial marine oligochaete proved "pollution-tolerant" in tests with an Arabian crude and new- and old-generation dispersants.[260]

Field Studies. Oil and intertidal sediments were investigated for changes during the year following the Amoco Cadiz spill for changes resulting from evaporation and degradation.[261] Studies of effects in biological systems included mollusk development,[262] changes in meiofauna diversity,[263] distribution of platyhelminthes,[264] community structure of *Zostera* beds,[265] and the occurrence of marine yeasts in degraded oil.[266]

Other catastrophic spills received interest through studies in the chemical similarities of oil weathering after release from offshore oil rigs,[267] population structure of littoral algae and invertebrates following the *Betelgeuse* explosion,[268] and zooplankton biomass distribution after the *Isesis* oil spill.[269] The chemical fate and intertidal biological consequences of the *Eleni V* spill were documented by Blackman and Law.[270] Research surrounding the *Florida* release off Falmouth harbor was detailed by Sanders *et al.*[271]

Documentation of inputs and effects of chronic oil release was made for diverse geographic areas including the Persian Gulf,[272] North Sea,[273] Coal Oil Point, California,[274] the benthos surrounding a deep-water production platform in the Gulf of Mexico,[275] and South African sandy beaches.[276] An examination of the effects of weathered and unweathered Louisiana crude oil experimentally spilled into Virginia estuaries was conducted by Bender *et al.*[277] Techniques for the evaluation of small gasoline spills, utilizing samples and bivalve mollusks, were detailed.[278]

Bioaccumulation. Research on the bioaccumulation of petroleum hydrocarbons was chronicled wherein clams were exposed to mixtures of hydrocarbons, usually as whole oil or derivatives.[279-281] Eight months after the *Amoco Cadiz* wreck, hydrocarbons accumulated in biogenic lipid-rich tissues of invertebrates and eels had been depurated to varying degrees.[282] Experimental studies of oil ingestion by eels were also reported.[283-284] Natural marine sediments were shown to accumulate significant levels of oil via suspended particulate settlement.[285] The marine diatom *Cyclotella* preferentially accumulated C-16 from a water and oil mixture.[286]

Accumulation kinetics were reported for specific aromatic hydrocarbons and marine organisms, such as the monoaromatic benzene and a rotifer.[287] The uptake of a diaromatic, naphthalene, by the polychaete *Arenicola* was described.[288] Palmork and Solbakken[289-290] showed that lobsters eliminated most accumulated phenanthrenes within a few weeks following a single dose and that maximal levels of hydrocarbon and metabolites in the liver of cartilaginous fishes occurred several days later than corresponding values in bony fish.

Dimethynaphtholene induced increases in the activity of microsonal NADPH neotetrazolium reductase in blood cells of *Mytilus edulis*.[291] The same species was able to metabolize 1-(1-^{14}C) napthol into 1-naphthylsulfate.[292] Anthracene caused damage to lysomes of *Mytilus* pericardial epithelium cells, implicating disruption of excretory processes involving blood purification.[293] Little differences in benzo(a)pyrene metabolism potential was noted between male and female skates.[294] Inhibition of oryl hydrocarbon hydroxylose in the gut of blue crabs by low concentrations of 7,8-naphtho-flavone and lack of induction by hydrocarbon-laced food suggested important differences between crab systems and those of other animals.[295]

Toxicity and Sublethal Effects. Emulsed oil collected some distance from the *Ixtoc I* runaway well was not acutely toxic to juvenile amphipods and zooplankton in the laboratory.[296] The acute toxicity of a fuel oil, wicking agent, and lighter fluid to an alga, copepod, and fish was reported.[297] Dibenzothiophene and 3,4-benzopyrene had no effect on the hatchability of *Artemia* eggs at levels up to 10 mg/l.[298]

Through studies on burrowing rate, respiration and growth suppression, it was concluded by Stekoll *et al.*,[299] that *Macoma balthica* would suffer population decreases when chronically exposed to oil-in-seawater concentrations exceeding 0.03 mg/l. Decreased spleen weight and increased hematocrit in females were the major effects resulting from exposure of fish to surface slicks for 2 weeks.[300] Petroleum hydrocarbons did not disrupt the intestinal mucosa cells of juvenile salmon at levels where PCBs were disruptive.[301] North Sea crude oil and several dispersants greatly increased the number of lysosomes in apical surfaces of gill cells from a gastropod.[302] Naphtholene proved more inhibitory than benzene to photosynthesis in macroalgae.[303] Higher doses of No. 2 fuel oil produced dose-response relationships for plasma corticosteroid, glucose and cholesterol levels in juvenile mullet.[304] Three studies characterized effects of petroleum hydrocarbons on osmotic physiology of mud crabs,[305] polychaetes,[306] and mummichog.[307]

Several papers described the disruption of growth and reproduction processes in marine organisms by petroleum hydrocarbons, primarily in the form of exposures to crude oils, including reduction of copepod brood size,[308] arrest of mussel growth at the hinged larvae stage,[309] modification of growth and predation rates for the oyster drill,[310] inhibition of sexual chemotoxis in *Fucus* spermatozoids,[311] mitochondrial swelling in herring embryonic muscle cells,[312] growth reduction in five phytoplankters[313] and growth inhibition in crab larvae.[314-315] Seasonal changes in effects of dispersant and fuel oil combinations on fin regeration were noted by Fingermann.[316] Behavioral work included quantitation of feeding rate disruption in *Acartia* copepod species and the determination of a threshold concentration of 10^{-7} mg/l for naphthalene in blue crab.[318]

Aves. Based on release and rehabilitation rates for a South African program, Randall *et al.*,[319] recommended attempting to clean and rehabilitate penguins after accidental oiling. Guidelines for evaluating effects of ingested oil on nesting seabirds were given.[320] McKelvey *et al.*,[321] called attention to the greater loss of seabirds around Vancouver harbor resulting from chronic vegetable-oil pollution as opposed to petroleum-oil pollution. In Arctic and sub-Arctic areas, light oilings can lead to avian mortality when oil effects synergistically combine with stresses imposed by severe climate.[322] Methods for the analysis of 16 hydrocarbons in avian tissue were detailed.[323] Incubation behavior of female ducks exposed to Prudhoe Bay crude oil (100 mg/m^2) was unaffected as was survival of first-week ducklings.[324] Effects of fresh and weathered crude oil hatchability of mallard, heron, and gull eggs were reported.[325-326]

Diseases and Abnormalities. Diseased populations of fish were reported in Massachusetts, New York, Southern California, South Africa, France, Tunisia and Japan,[327-328] Puget Sound,[329] the German Bight,[330] the Northeast Atlantic Ocean, and Northeast Pacific Ocean.[330]

Peters and Watermann[330] recognized three types of skin tumors in near-shore marine fish:

- infectious "fish pox"-like tumors affecting up to 6.6% of Atlantic dab (*Limanda limanda*) possibly related to sulfuric acid and titanium contamination in the German Bight;
- lip papillomas caused by direct mechanical abrasion in many captive fish; and
- epidermal papillomas in pleuronectid flatfish of the North Pacific that may be caused by a virus and may be pollution-related. However, Dawe *et al.*, concluded these latter types of tumors were caused by an amoeba; these authors and Hard *et al.* agreed they are not neoplastic, as once presumed.

Hepatomas (liver tumors) and several pre-cancerous liver lesions have been reported in 25% of Atlantic tomcod (*Microgadus tomcod*) from the Hudson River estuary[333] and lower frequencies in several species of demersal fish in contaminated waterways of Puget Sound, Washington.[329] The Hudson River hepatocellular carcinomas are associated with fish containing PCBs in the range of 10 to 90 μg/l. Several distinct types of liver lesions in Puget Sound flatfish occurred in fish contaminated with PCBs, chlorinated pesticides, chlorinated butadienes and petroleum hydrocarbons; however, the causes of these disorders have yet to be confirmed. Mutagenic chemicals were also detected in British mussels.[335]

Malins *et al.*[329] also reported a variety of lesions in shrimp and crabs taken from contaminated waterways of Puget Sound while another disease was characterized for shrimp from the Gulf of Mexico.[336] Zinc was found to accelerate regenerative growth of cut fins in *Fundulus* exposed to methyl mercury; in the absence of zinc, methyl mercury inhibited regeneration.[337]

Sludge, Dredging, Ocean Dumping, and Recovery. The composition, fate and effects of waste activated sludge disposal by

the City of Los Angeles into Santa Monica Bay was reviewed and brought up to date with new samplings.[328] About 10% of sludge discharged to date remained on the sea bottom covering an area of 3 km^2 of sea bottom or 0.4% of the Bay. Densities of bottom fish, crustaceans and infauna were increased at the site while diversity of infauna was depressed. Several trace elements were elevated in invertebrates but were not biomagnified through the food web. In contrast, PCBs were elevated in sediments, fish and shellfish, but most organisms met the FDA standard. Sludge dumping in the New York Bight has elevated levels of PCBs in adjacent sediments, but contamination was limited to the dump site, and dated core profiles showed good agreement with initiation of production and use.[337] However, the presence of coprostanol, a fecal steroid, indicated sediments of the New York Bight are probably more widely contaminated with barge-dumped wastewater sludge.[342] Wastewater sludge disposal in tropical mangroves resulted to accumulation of Cd, Cu, Cr, Zn, Ni, and Pb in turtle grass.[343] Swan[344] reported on NOAA studies on the distribution, fate and effects of pharmaceutical sludges dumped at sea over the Puerto Rico Trench. Atlas *et al.*,[345] reported that a biological sludge dumped by a chemical company at a dumpsite in the Gulf of Mexico contained over 100 organic compounds and metals. Phytoplankton growth was enhanced slightly but did not lead to other measurable effects, such as occurrence of chemical residues or decreased transparency 24 hours after dumping. At a 2 800-m deep Atlantic dumpsite, canister-contained radio-cesium (^{137}Cs) leaked into adjacent water and sediments.[346] Observed ^{137}Cs concentrations in sediments were several orders of magnitude above background and bioturbation seemed to be the dominant mechanism for dispersal.

Recovery of macrobenthic communities began about 6 months after dredging a section of Intracoastal Waterway in South Carolina; to minimize adverse effects the authors recommended dredging in late fall/early winter, dredging more frequently and disposing at several sites.[347] Brannon and Plumb[348] reported no long-term beaching problems would be associated with disposal of 32 types of dredged materials. Bohlen *et al.*[349] reported the distribution and fate of suspended matter during estuarine-channel dredging operations; from 1.5 to 3.0% of each bucket volume was lost in the water column. No evidence was found for the biomagnification of trace elements in marine food webs associated with contaminated outfall and harbor sediments; several metals and benzo(a)pyrene actually decreased with increased trophic level. Trophic levels were identified using both feeding-habit data and the cesium/potassium ratio, a possible index of trophic level.[350-351]

Data from a series of related studies[338-341] confirmed that significant recovery of kelp beds, intertidal seaweeds, benthic crustaceans, and bottom fish population has taken place during the past several years near the large Palos Verdes wastewater outfall in southern California. Reduction in solids emissions, kelp restoration, and source control of toxic chemicals and advantageous oceanographic episodes have all contributed to the recovery. Effects such as fin erosion and occurrence of pollution tolerant macrofaunal organisms continue, however.

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